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Landscape variation in tree regeneration and snag fall drive fuel loads in 24-year old post-fire lodgepole pine forests

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Abstract. Escalating wildfire in subalpine forests with stand-replacing fire regimes is increasing the extent of early-seral forests throughout the western USA. Post-fire succession generates the fuel for future fires, but little is known about fuel loads and their variability in young post-fire stands. We sampled fuel profiles in 24-year-old post-fire lodgepole pine (*Pinus contorta* var. *latifolia*) stands (n = 82) that regenerated from the 1988 Yellowstone Fires to answer three questions. (1) How do canopy and surface fuel loads vary within and among young lodgepole pine stands? (2) How do canopy and surface fuels vary with pre- and post-fire lodgepole pine stand structure and environmental conditions? (3) How have surface fuels changed between eight and 24 years post-fire? Fuel complexes varied tremendously across the landscape despite having regenerated from the same fires. Available canopy fuel loads and canopy bulk density averaged 8.5 Mg/ha (range 0.0–46.6) and 0.24 kg/m^3 (range: 0.0–2.3), respectively, meeting or exceeding levels in mature lodgepole pine forests. Total surface-fuel loads averaged 123 Mg/ha (range: 43–207), and 88% was in the 1,000-h fuel class. Litter, 1-h, and 10-h surface fuel loads were lower than reported for mature lodgepole pine forests, and 1,000-h fuel loads were similar or greater. Among-plot variation was greater in canopy fuels than surface fuels, and within-plot variation was greater than among-plot variation for nearly all fuels. Post-fire lodgepole pine density was the strongest positive predictor of canopy and fine surface fuel loads. Pre-fire successional stage was the best predictor of 100-h and 1,000-h fuel loads in the post-fire stands and strongly influenced the size and proportion of sound logs (greater when late successional stands had burned) and rotten logs (greater when early successional stands had burned). Our data suggest that 76% of the young post-fire lodgepole pine forests have 1,000-h fuel loads that exceed levels associated with high-severity surface fire potential, and 63% exceed levels associated with active crown fire potential. Fire rotations in Yellowstone National Park are predicted to shorten to a few decades and this prediction cannot be ruled out by a lack of fuels to carry repeated fires.

Key words: fire regimes; fuel dynamics; fuels; lodgepole pine; Pinus contorta *var.* latifolia; *reburn; self-regulation; succession; Yellowstone National Park; young forests.*

Introduction

Observed and projected increases in wildland fire extent and frequency have raised concern among scientists and forest managers regarding the consequences of escalating wildland fire activity (Flannigan et al. 2000, Scholze et al. 2006, Westerling et al. 2006, 2011, Krawchuk et al. 2009, Moritz et al. 2012, Stephens et al. 2013, Parks et al. 2015). Extreme fire seasons have become more common over the last three decades and have had major social and ecological consequences, including loss of human life and infrastructure, escalating costs of fire prevention and suppression, changes in ecosystem services (e.g., water, timber, carbon storage, and recreation resources), and increasing extents of young forests (Schoennagel et al. 2006, Stephens et al. 2013). In subalpine forests across western North America, large fires historically burned during rare periods of extreme weather (Romme 1982, Lotan et al. 1985, Bessie and Johnson 1995, Schoennagel et al. 2004). Projections of more frequent severe fire weather over longer fire seasons suggest that a new wildland fire issue may emerge: the potential for extensive reburning of young forests (Schoennagel et al. 2006, Parks et al. 2015, 2016, Harvey et al. 2016). If realized, forest managers and scientists will be challenged to anticipate successional dynamics that ultimately generate the fuels for future fires. In this study, we evaluate patterns of fuel accumulation and abundance in young, post-fire lodgepole pine (*Pinus contorta* var. *latifolia*) stands originating from the 1988 Yellowstone Fires to understand variation in fuel loads in young, post-fire forests.

Observations of natural fires in Yellowstone during the 1970s and 1980s suggested that young (≤40 yr) post-fire lodgepole pine were unlikely to burn because combustion from the first fire had reduced fuel loadings (Renkin and Despain 1992). The spread of fires that burned early

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during the 1988 fire season fit that expectation, slowing when patches of young forest were encountered (M. G. Turner and W. H. Romme, *personal observations*). However, fires that burned later in the 1988 fire season under extreme weather conditions burned readily through young lodgepole pine forests at seven (1981 Pelican Creek Fire) and 13 (1975 Arrow Fire) years post-fire. Recent fires in the Greater Yellowstone Ecosystem have reburned lodgepole pine stands at 12 (2000 Boundary fire), 24 (2012 Cygnet Fire), and 28 (2009 Bearpaw Fire) years post-fire. Parks et al. (2015) rigorously tested the ability of young forests to act as fire breaks and found that the likelihood of re-burning was reduced for 14–18 years in four northern Rocky Mountain forest landscapes. Harvey et al. (2016) also conducted rigorous sampling and analysis in northern Rocky Mountain forest landscapes and found that burn severity was reduced for 10–12 years in subalpine forests, but a second fire after that time was likely to be high severity. Both studies found that extreme burning conditions could negate any reduced likelihood of fire or burn severity in young forests.

Fuel dynamics following fire thus govern the likelihood that young stands will again burn (Parks et al. 2015, Harvey et al. 2016), but comprehensive measurements of post-fire fuel loads and variability in young stands are lacking. Previous fuel succession studies indicate that post-fire fuel loads capable of supporting fire vary by forest type. Western hemlock/Douglas-fir forests in Washington, USA (*Tsuga heterophylla*/*Pseudotsuga menziesii*; Agee and Huff 1987), and subalpine fir forests in Montana, USA (*Abies lasiocarpa*; Fahnestock 1976), show high, post-fire fuel loads and potential for short-interval fire; however, lodgepole pine forests in Wyoming, USA (Romme 1982, Tinker and Knight 2000), and Scots pine/Norway spruce/birch forests in northern Sweden (*Pinus sylvestris*/*Picea abies*/*Betula* spp.; Schimmel and Granström 1997) appeared to require extended periods of biomass accumulation to support subsequent fire. The objective of this study was to quantify fuel loads and variability at the landscape scale following a large, severe wildfire. The 1988 Fires in Yellowstone National Park provide an ideal opportunity for such a study because the post-fire forest landscape has received minimal human intervention and the consequences of the fires have been studied extensively (e.g., Turner et al. 2003, Turner 2010, Romme et al. 2011).

After nearly 25 years of succession following high-severity fire, young lodgepole pine stands in Yellowstone National Park vary widely in structure and function (Turner et al. 2016), and have developed complex and varied fuel profiles. We sampled fuels across the forests regenerating from the 1988 fires to answer three questions: (1) How do canopy and surface fuel loads vary within and among young lodgepole pine stands across the burned landscape? (2) How do canopy and surface fuels vary with pre- and post-fire lodgepole pine stand structure and environmental conditions? (3) How have surface fuels changed between eight and 24 years after the 1988 Yellowstone Fires?

Study Area

The subalpine plateau of Yellowstone National Park is a mostly roadless landscape dominated by lodgepole pine but also contains Engelmann spruce (*Picea engelmanii*), subalpine fir, and whitebark pine (*Pinus albicalus*) in lower numbers. Most of the park ranges between 2,100 and 2,700 m in elevation. Soils include dry, infertile, rhyolitic substrates as well as more mesic and slightly less infertile andesitic and former lake-bottom substrates (Turner et al. 2004). The hydrologic regime is dominated by winter snowfall which generally persists from late-October to late-May at 2,100 m elevation (Despain 1990). Between 1981 and 2010 at Old Faithful, mean annual temperature was 1.2°C, ranging from an average low of −17.6°C in January to an average high of 23.8°C in July (*data available online*).[5] Mean annual precipitation ranges from 366 to 642 mm depending on geographic location and elevation (see footnote 5).

Wildland fires within Yellowstone National Park generally ignite from lightning associated with convective summer storms and the 1988 fires did so during the warmest, driest summer on record (Renkin and Despain 1992). These fires produced an extremely heterogeneous mosaic of burn severities across ~600,000 ha within the Greater Yellowstone Ecosystem (Christensen et al. 1989). Of the ~321,000 ha that burned inside the park, ~56% burned as stand-replacing fire, with 25% classified as severe-surface burn and 31% as high-severity crown fire (Turner et al. 1994). Post-fire lodgepole pine regeneration was rapid, abundant, and remarkably variable across the burned landscape, with post-fire tree densities ranging from 0 to >500,000 stems/ha (Turner et al. 1997, 1999, 2004, Turner 2010). Large stand-replacing fires historically occurred at a 100–300 yr interval (Romme 1982, Millspaugh and Whitlock 1995), but scientists and managers alike found the 1988 fires to be surprising and noteworthy in fire extent, severity, and rate of forest recovery (Romme et al. 2011). Extreme fire weather in the study area is projected to become more frequent and longer in duration over the next century leading to a reduction in fire rotation from >100 to <30 yr (Westerling et al. 2011).

Methods

During the summer of 2012, we measured canopy, surface, and herbaceous fuels in 24-yr-old post-fire forests across Yellowstone National Park. Ten plots were originally established in 1996 (Tinker and Knight 2000) and 72 plots were established in 1999/2000 (Turner et al. 2004). Plots encompassed a wide range of post-fire stem density, two 1988 stand-replacing fire-severity categories (i.e., crown and high-severity surface fire), and four substrate categories: rhyolite, till; rhyolite, glacial; rhyolite, low base saturation; and andesite includes lake bottom sediments. Surface fuels had previously been measured in

[5] http://www.wrcc.dri.edu/

1996 in 10 plots (Tinker and Knight 2000) but no fuel measurements were collected in the Turner et al. (2004) plots in 1999/2000. All 82 plots were used to evaluate our first two questions, and the 10 plots sampled in 1996 were re-measured to address our third question. Sampling locations were separated by at least 1 km and spatial independence was confirmed using the Moran's I test ($P = 0.192$). To our knowledge, this study includes one of only a few expansive fuels dataset collected within a single wildfire footprint.

Field measurements

2012 canopy, surface, and understory vegetation fuels.—We measured canopy, surface, and understory vegetation fuels at all 82 sites using a 0.25-ha (50 × 50 m) fixed-area plot. Plots were oriented northward with a southerly baseline that ran east–west (see Turner et al. 2004). Twelve 20-m planar intercept fuels transects were oriented within each plot (Brown 1974, Brown et al. 1982). One-h (<0.64 cm diameter) and 10-h fuels (0.64–2.54 cm diameter) were tallied along the first 3 m of each transect, 100-h fuels (2.54–7.62 cm diameter) were tallied along the first 10 m, and 1,000-h fuels (>7.62 cm diameter) were measured along the full 20 m. Litter depth (cm) was measured at three locations spaced at 2-m intervals at the beginning of each transect. Litter is defined as lightly decomposed recognizable organic matter and duff is defined as decomposed, unrecognizable organic matter. Duff was absent in all plots, which contrasts with boreal forests but is typical in young lodgepole pine forests.

We assessed canopy fuel profiles from estimates of stem density and size in each plot (Turner et al. 2016). Briefly, all trees were tallied by species inside three 2 × 50 m belt transects that ran through the center and along the east and west boundaries of the plot. On a sample of lodgepole pine trees ($n = 25$) in each plot, we measured basal diameter (0.1-cm), diameter at breast height (dbh = 1.37 m, 0.1 cm), tree height (0.1 m), crown base height (0.1 m), and crown width (0.1 m). Understory vegetation cover was estimated visually by species within 25 0.25-m^2 quadrats and converted to biomass using allometric equations developed in Yellowstone for these species (Turner et al. 2004, 2016, Simard et al. 2011).

We also measured litter bulk density (kg/m^3) in 14 plots that spanned a representative range of post-fire seedling densities. Twelve subsamples of litter were collected using a 0.30 × 0.30 m quadrat, and litter depth was recorded at the center of the quadrat. Litter was dried at 60°C for 24 h or until a constant mass was reached and weighed on an analytical balance to a hundredth of a gram. Care was taken to remove woody particles and mineral soil from each sample.

1996 surface fuels.—In 1996, surface fuels were measured in 10 post-fire plots (Tinker and Knight 2000). At each plot, 25 15.2-m transects were oriented at random azimuths on the east and west plot boundaries. Surface fuel loads were assessed along each transect using the planar intercept method (Brown 1974, Brown et al. 1982). One-h (<0.64 cm diameter) and 10-h fuels (0.64–2.54 cm diameter) were tallied along the first 1.83 m of each transect, 100-h fuels (2.54–7.62 cm diameter) were tallied along the first 3.66 m, and 1,000-h fuels (>7.62 cm diameter) were measured along the full 15.2 m. Litter depth (cm) was measured along each transect at three locations (0.67 m apart). Duff was absent in all plots.

Data processing

Surface fuel loads were computed for each plot by summing intercept counts and transect lengths by size class, applying standard planar-intercept methods (Brown 1974, Brown et al. 1982, Harmon et al. 1986), then scaling to the hectare. 1,000-h fuel loads were summarized and grouped into two classes, sound and rotten, depending on their decay status (Maser et al. 1979). Sound logs include those with a sound bole regardless of branch, bark, and twig condition (i.e., decay classes 1 and 2). Rotten logs include those with a bole that breaks apart (i.e., decay classes 3, 4, and 5). Percent cover by species was converted to understory vegetation biomass using published relationships between percent cover and dry biomass (Turner et al. 2004, Simard et al. 2011). Mean litter bulk density (kg/m^3) was computed using a subsample of plots ($n = 14$), and litter loads were computed for each plot by multiplying litter depth (m) by litter bulk density. Canopy fuel loads were calculated by applying custom lodgepole pine allometric equations (Copenhaver and Tinker 2014) to the randomly subsampled trees, taking their plot-wise mean, then scaling to the hectare with tree density. Foliage, 1-h branch wood, and available canopy fuel loads are reported. Available canopy fuel load is defined as the proportion of canopy fuels available for pyrolysis and is computed as 100% of foliage plus 50% of 1-h branch wood (Reinhardt et al. 2006). We define crown as pertaining to an individual tree and canopy as the sum of all individual trees within a stand (Cruz et al. 2003). Crown bulk density was computed using Eq. 1.

Canopy bulk density was computed using the biomass-percentile method (Reinhardt et al. 2006). Canopy length was determined by summing biomass through the canopy and reporting the distance between the 10th and 90th

$$\text{Available crown bulk density}\,(\text{kg m}^{-3}) = \frac{\text{foliage (kg)} + 0.5(\text{1 h branchwood (kg)})}{\pi\left(\frac{\text{crown width (m)}}{2}\right)^2 \times (\text{tree height (m)} - \text{crown base height (m)})} \tag{1}$$

Table 1. Fuel characteristics in low, moderate, and high density stands.

	Density class			
	Low (<1,000 stem/ha)	Moderate (1,000–50,000 stem/ha)	High (>50,000 stem/ha)	All
Sample size	n = 17	n = 56	n = 9	n = 82
Stand density (trees/ha)	430 (67)	8,771 (1,149)	124,474 (36,482)	19,508 (5,645)
Crown				
Mean crown base height (m)	0.14 (0.02)[a]	0.47 (0.03)[b]	0.69 (0.07)[c]	0.42 (0.03)
Crown bulk density (kg/m^3)	0.60 (0.07)[a]	0.75 (0.04)[a]	1.51 (0.17)[b]	0.80 (0.04)
Canopy				
Foliage biomass (Mg/ha)	0.84 (0.18)[a]	8.27 (0.73)[b]	15.12 (3.65)[c]	7.49 (0.76)
1-h branch biomass (Mg/ha)	0.26 (0.06)[a]	2.43 (0.21)[b]	3.81 (1.09)[b]	2.14 (0.22)
Available canopy fuel load (Mg/ha)†	0.97 (0.20)[a]	9.50 (0.84)[b]	16.64 (4.25)[c]	8.53 (0.87)
Total canopy biomass (Mg/ha)	3.52 (0.74)[a]	34.94 (3.11)[b]	63.73 (15.62)[c]	31.63 (3.25)
Canopy length (m)	3.48 (0.34)[a]	3.77 (0.12)[a]	1.77 (0.32)[b]	3.49 (0.13)
Canopy bulk density (kg/m^3)‡	0.03 (0.00)[a]	0.24 (0.02)[b]	0.66 (0.12)[c]	0.24 (0.03)
Live surface fuels				
Herbaceous (Mg/ha)	1.71 (0.10)[a]	0.98 (0.09)[b]	0.81 (0.15)[b]	1.11 (0.07)
Shrub (Mg/ha)	0.10 (0.03)[a]	0.15 (0.02)[a]	0.13 (0.05)[a]	0.13 (0.02)
Dead surface fuels				
Litter depth (cm)	0.59 (0.16)[a]	1.19 (0.10)[b]	1.73 (0.36)[b]	1.10 (0.09)
Litter (Mg/ha)	2.98 (0.78)[a]	5.96 (0.51)[b]	8.70 (1.80)[b]	5.61 (0.46)
1-h (Mg/ha)	0.10 (0.01)[a]	0.18 (0.02)[b]	0.29 (0.04)[c]	0.17 (0.01)
10-h (Mg/ha)	2.02 (0.19)[a]	2.35 (0.11)[a]	2.35 (0.33)[a]	2.28 (0.09)
100-h (Mg/ha)	4.45 (0.32)[a]	4.95 (0.23)[a]	7.18 (1.13)[b]	5.08 (0.22)
Sound 1,000-h (Mg/ha)	78.48 (7.54)[a]	82.52 (5.64)[a]	53.96 (7.83)[a]	78.55 (4.31)
Rotten 1,000-h (Mg/ha)	39.24 (4.84)[a]	31.13 (2.68)[a,b]	17.64 (5.33)[b]	31.42 (2.23)
Total surface fuel load (Mg/ha)	127.27 (10.62)[a]	127.09 (5.62)[a]	90.12 (10.33)[b]	123.12 (4.70)

Notes: Mean ± 1 SE. Letters indicate row-wise differences (Tukey's HSD, α = 0.05).
†Available canopy fuel load = foliage + 0.5 × (1-h branch wood).
‡Computed using the biomass-percentile method (Reinhardt et al. 2006).

percentile of biomass. Vertical fuel profiles were created for each plot by splitting available canopy fuel load into 0.1-m vertical layers then dividing by the volume of each layer (plot area × layer depth; Sando and Wick 1972, Reinhardt et al. 2006). For plotting purposes, a 1-m running mean was used to smooth the fuel profile and reduce extreme values. Within-plot coefficient of variation (CV) was computed for each plot by computing fuel loads for each measurement unit (i.e., transects, n = 12 per plot, and trees, n = 25 per plot) and aggregating these estimates into plot-level means, standard deviations, and CVs. Among-plot (i.e., landscape-wide) CVs were also computed for each fuel category using among-plot means and standard deviations (n = 82).

Statistical analyses

To examine fuel loads and their variability, we report means and standard errors of plot-level fuel loads (Table 1) and within-plot and among-plot coefficient of variation (Table 2) for three tree density classes and the total population. Differences among density classes were determined using one-way ANOVA and Tukey's HSD at α = 0.05. Landscape-wide CVs pertain to the whole study area and lack error estimates. We also calculated the proportion of the 82 plots that met or exceeded fuel loads associated with the potential for high-severity surface fire (1,000-h fuel loads greater than 60 Mg/ha; Sikkink and Keane 2012) and active or independent crown-fire spread (crown bulk density greater than 0.12 kg/m^3; Reinhardt et al. 2006).

To assess how fuel loads varied with pre- and post-fire stand structure and topo-climatic factors, we fit linear multiple regression models to predict fuel loads. Candidate predictor variables were selected based on the hypothesized ecological relationships and paired to each response variable (Appendix S1: Table S1). Models were selected using the best subsets model selection routine to optimize the coefficient of determination (R^2) while maintaining α = 0.05 (Lumley and Miller 2009). Model residuals, fits, and transformation criteria were checked using methods recommended by Venables and Ripley (2002), and the presence of multicollinearity was evaluated using variance inflation factors. After model construction, the importance of individual predictor variables was evaluated by computing the proportion of R^2 that each predictor variable contributes using the lmg metric in the relaimpo R package (Grömping 2006). lmg "quantifies the relative contributions of the regressors to the model's total explanatory value by averaging sequential sums of squares over orderings of regressors" (Grömping 2006, 2007). Significant differences in lmg between predictor variables were tested by generating bootstrapped confidence intervals (999 iterations).

Empirical response variables and transformations include litter ($\log_{10}$), 1-h ($\log_{10}$), 10-h, 100-h ($\log_{10}$), 1,000-h

Table 2. Within- and among-plot variability of fuel loads and fire behavior parameters in low, moderate, and high density stands.

	Within-plot variation (CV) by stem-density class				
	Low (<1,000 stems/ha)	Moderate (1,000–50,000 stems/ha)	High (>50,000 stems/ha)	Within all plots	Among-plot variation
Sample size	$n = 17$	$n = 56$	$n = 9$	$n = 82$	$n = 82$
Crown					
Mean crown base height	113.1 (26.7)[a]	70.8 (3.9)[b]	45.3 (4.8)[b]	76.7 (3.4)	65.5
Available crown bulk density	116.8 (26.9)[a]	103.2 (5.4)[a]	93.9 (7.9)[a]	105.0 (6.6)	112.4
Canopy					
Foliage biomass	123.4 (25.3)[a]	113.8 (3.8)[a]	97.4 (11.6)[a]	114.0 (5.9)	96.3
1-h branch biomass	128.8 (25.2)[a]	120.6 (4.0)[a]	105.2 (12.6)[a]	120.6 (5.9)	96.1
Available canopy fuel load	124.1 (25.3)[a]	114.7 (3.9)[a]	98.5 (11.8)[a]	114.9 (5.9)	96.2
Total canopy biomass	123.1 (25.3)[a]	113.4 (3.8)[a]	97.0 (11.5)[a]	113.6 (5.9)	96.1
Dead surface fuels					
Litter	131.7 (17.5)[a]	97.2 (5.7)[b]	75.8 (4.8)[b]	101.9 (5.6)	74.4
1-h	89.5 (7.0)[a]	85.7 (4.3)[a]	65.7 (6.2)[a]	84.3 (3.4)	70.4
10-h	84.0 (7.4)[a]	74.2 (3.1)[a]	80.2 (10.6)[a]	76.9 (2.9)	36.5
100-h	59.1 (3.3)[a]	60.3 (3.5)[a]	68.3 (5.0)[a]	60.9 (2.6)	39.8
Sound 1,000-h	51.0 (3.8)[a]	47.8 (6.0)[a]	52.8 (4.0)[a]	49.0 (1.9)	49.9
Rotten 1,000-h	68.9 (4.1)[a]	67.5 (4.0)[a]	53.0 (8.8)[a]	66.2 (3.1)	64.9
Total surface fuel load	39.5 (2.6)[a]	35.5 (1.5)[a]	34.5 (2.3)[a]	36.2 (1.2)	34.9

Notes: Within-plot variability estimates are mean coefficient of variation (CV, percentage) ± 1 SE. Letters indicate row-wise differences (Tukey's HSD, $\alpha = 0.05$). Among-plot coefficient of variation was computed using the population standard deviation and mean and do not include error rates.

rotten ($\log_{10}$), 1,000-h sound ($\log_{10}$), total surface fuel, live herbacious biomass, live shrub biomass ($\log_{10}$), available canopy fuel load ($\log_{10}$), mean crown base height ($\log_{10}$), and canopy bulk density ($\log_{10}$). Empirical predictor variables and transformations include live stem density ($\log_{10}$), live foliage, herbaceous, shrub biomass, live tree density, live basal area, canopy base height, and 1,000-h fuel load. Geospatial predictor variables were extracted from the following datasets using plot coordinates: pre-fire successional stage (Despain 1990, NPS-YELL 1990), substrate (NPS-YELL 1997), mean annual precipitation and temperature (PRISM Climate Group 2012), and burn severity (dNBR; USDA Forest Service-RSAC 2012). The following geomorphometric predictor variables were computed using a digital elevation model (Gesch 2007) and extracted using plot coordinates: derived slope and aspect (Gesch 2007), compound topographic index (i.e., wetness; Evans et al. 2014), and potential solar radiation (Pierce et al. 2005). Aspect was transformed to a continuous distribution using Beers et al. (1966). Categorical dummy variables, pre-fire successional stage and substrate, were defined in our models using deviance (effects) contrasts to compare individual levels with the mean of all levels. Pre-fire successional stage reflects lodgepole pine successional stages identified by Despain (1990). LP0 represents post-fire stands where lodgepole pine has recolonized the site but has not yet produced a closed canopy. LP1 consists of a single cohort of dense, young lodgepole pine without tree seedlings in the understory. LP2 stands have closed canopies dominated by lodgepole pine with tree seedlings in the understory. LP3 and LP4 are multi-cohort stands with ragged canopy characteristics dominated by lodgepole pine. LP3 contains Engelmann spruce and subalpine fir in the sub-canopy whereas LP4 stands occur on dry sites that do not support Engelmann spruce and subalpine fir.

Total crown area was modeled using linear regression between summed tree crown area and stand density. Changes in fuel loads and their variability between 1996 and 2012 were compared using paired *t* tests and the ratio of change was calculated using the ratio of means for each surface fuel type plus standard error (Scheaffer et al. 2011).

Results

Fuel characteristics and variability

Total surface fuel loads varied tremendously across the post-1988 Yellowstone fire landscape, ranging from 43.3 to 206.7 Mg/ha (Fig. 1, Table 1). 1,000-h fuels averaged 110.0 ± 4.6 Mg/ha and composed 88% of the total fuel, by far the greatest proportion. Sound logs accounted for nearly 70%, while rotten logs accounted for 30% of 1,000-h biomass but the distribution of biomass also varied with log size (Fig. 2). 75.9% of stands had 1,000-h fuel loads greater than 65 Mg/ha, a threshold specified by Sikkink and Keane (2012) for high-severity surface fire. Litter accounted for the greatest share of fine surface fuels, more than 1-h surface fuels, herbaceous fuels, shrub fuels, and 10-h fuels combined. Mean litter bulk density was 50.2 ± 6.1 kg/m^3, mean litter depth was 1.1 ± 0.1 cm, and mean litter biomass was 5.61 ± 0.46 Mg/ha. Fuel loads increased with density class for litter, 1-h, and 100-h fuel types but decreased for herbaceous biomass, rotten 1,000-h and total surface fuel loads (Table 1). Litter and rotten 1,000-h fuel within-plot variation decreased by density class but other fuel classes did not. Across all

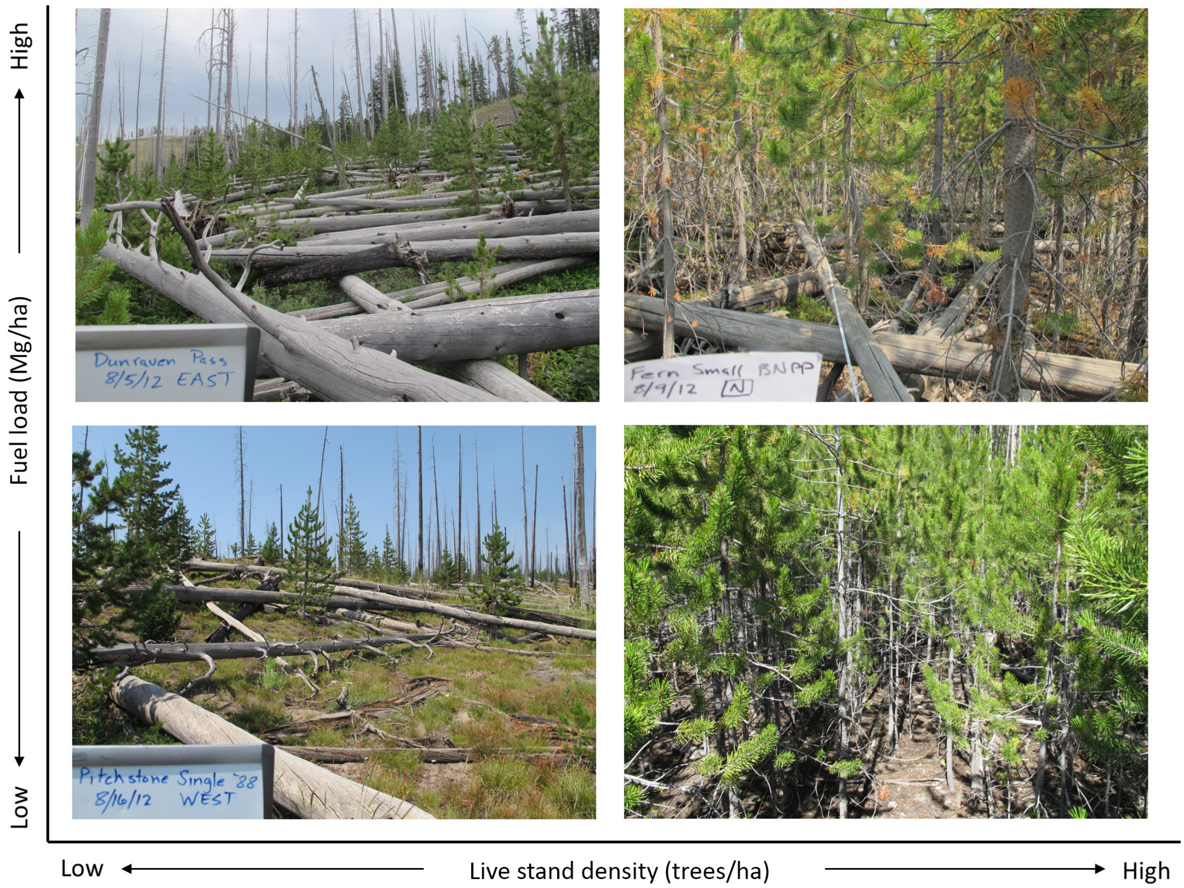

FIG. 1. The great range of variability in 24-yr-old lodgepole pine forest structure and fuel characteristics developing after the 1988 Yellowstone fires. Photos taken in 2012 by K. N. Nelson and M. G. Turner.

density classes, mean within-plot CVs were higher than landscape-wide CVs for the same fuel type (Table 2). In general, at the within- and among-plot scales, surface fuel variability was less than canopy fuel variability.

Live lodgepole pine densities in 2012 averaged 19,500 stems/ha and ranged from 0 to 344,000 trees/ha (Turner et al. 2016). Available canopy fuels averaged 8.5 Mg/ha and varied from 0.0 to 48.6 Mg/ha over this wide range of stem density (Fig. 1, Table 1). All canopy fuel characteristics (i.e., foliar biomass, 1-h biomass, crown base height) increased with stem density. Within-plot CV for canopy fuels did not differ among density classes but was greater than the among-plot CV for canopy fuels (Table 2). Canopy bulk density ranged from 0.00 to 2.28, and 63.9% of stands in this study are greater than the 0.12 kg/m^3 threshold for active and independent crown fire spread (Reinhardt et al. 2006).

Effects of pre- and post-fire stand conditions and topo-climatic factors on fuels

Models predicting dead and downed surface fuel loads fell into two general groups: fine fuels (e.g., litter, 1-h, and 10-h fuels) were best predicted by post-fire stand structure, and coarse fuels (e.g., 100-h, 1,000-h) were best predicted by pre-fire forest structure variables (Table 3). Needle litter fuel load was positively associated with 1,000-h fuel load (both sound and rotten) and live stand density but was negatively associated with mean annual precipitation (R^2 = 0.43). Live stem density showed the greatest importance in predicting litter fuel loads (Table 5). One-h fuel load was positively related to crown base height, mean annual precipitation, topographic wetness index, and slope (R^2 = 0.49) with crown base height having greater importance than the other variables in our model (Table 5). Variation in lodgepole pine density and annual temperature were related to 10-h fuels but explained little variance (R^2 = 0.08). 100-h fuels were best predicted by pre-fire successional stage and annual precipitation (R^2 = 0.22). Sound 1,000-h fuels were positively related to pre-fire successional stage (R^2 = 0.24), whereas rotten 1,000-h fuels were negatively related to pre-fire successional stage (R^2 = 0.30; Table 3, Fig. 2). Total surface fuel loads were best predicted by pre-fire successional stage, mean annual temperature, mean annual precipitation, and aspect (R^2 = 0.31). Herbaceous fuel load was best predicted by soil class and live tree basal area (R^2 = 0.25; Table 4).

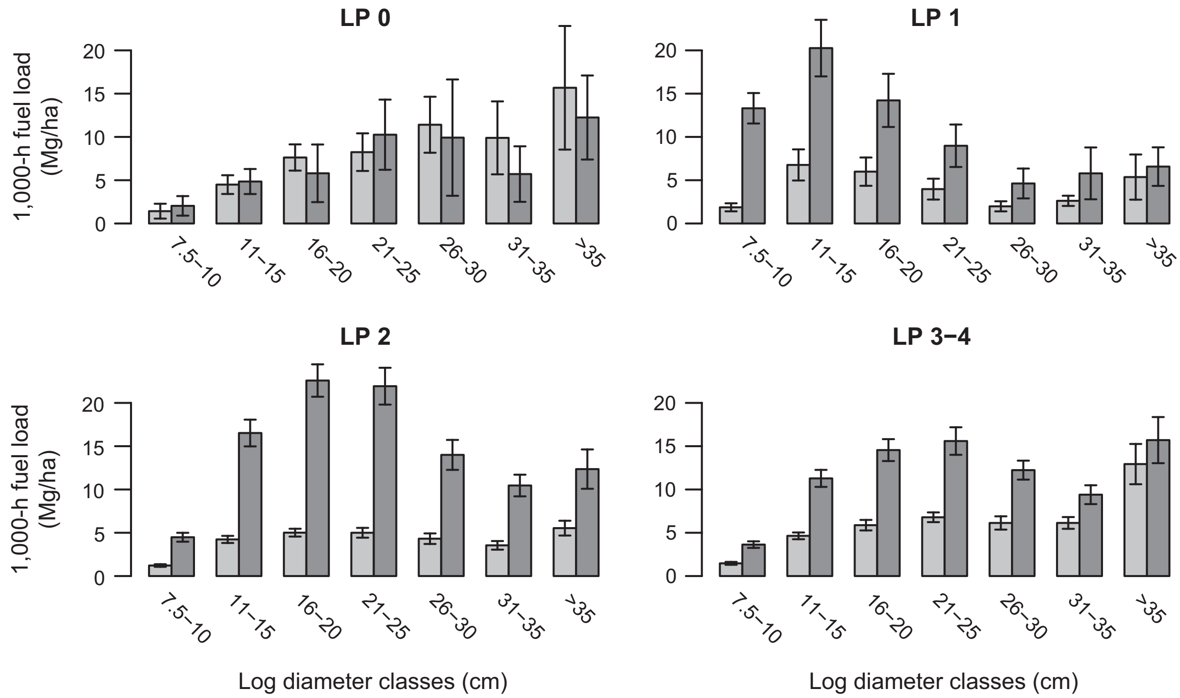

Fig. 2. Coarse fuel loads by log size and decay status for each pre-fire successional stage. Rotten log fuel loads are depicted with light gray bars and sound log fuel loads are depicted with dark gray bars. Pre-fire vegetation successional stages (Despain 1990) include LP0, post-fire stands where lodgepole pine has recolonized the site but has not yet produced a closed canopy; LP1, dense, young lodgepole pine in a single cohort without tree seedlings in the understory; LP2, closed canopy lodgepole pine with tree seedlings in the understory; LP3, multi-cohort stands with ragged canopy characteristics dominated by lodgepole pine but containing Engelmann spruce and subalpine fir in the sub-canopy; LP4, seral lodgepole pine stands on dry sites wtihout Engelmann spruce and subalpine fir. LP3 and LP4 have similar aboveground biomass characteristics and were combined to enhance sample size.

Shrub fuels were miniscule but were positively related to live stand density, elevation, and slope ($R^2 = 0.25$; Table 4). Importance values for predictors were not different from one another in 10-h, 100-h, 1,000-h, total fuel load, herbaceous, and shrub models (Table 5).

Live stem density was a strong, positive predictor of available canopy fuel (R^2 = 0.78), canopy bulk density (R^2 = 0.87), and crown base height (R^2 = 0.66; Table 4) and had the greatest importance in predicting canopy fuel loads (Table 5). Vertical profiles of canopy bulk density display a shift in the distribution of canopy fuels with stem density (Fig. 3). Low-density stands have lower canopy bulk density and a uniform vertical distribution whereas high-density stands had high canopy bulk density concentrated at lower heights. Crown area increased with stand density (R^2 = 0.80) and equaled ground area at ~12,000 trees/ha (Fig. 4).

Change in surface fuel loads with time since fire

Surface down and dead fuel loads in post-1988 wildfire forests generally increased with time since fire; however, 1-h fuels did not change between 1996 and 2012 (Table 6). Rotten 1,000-h fuels increased by a factor of four; other fuel classes increased by approximately half that rate (Table 6). Increasing fuel loads coincided with a sharp decrease in within-plot variability (Fig. 5). 1,000-h fuel loads greater than the 65 Mg/ha threshold for severe surface fire (Sikkink and Keane 2012) were present in 10% of stands in 1996 and 90% of stands in 2012.

Discussion

A quarter-century after the 1988 fires in Yellowstone National Park, substantial fuel loads cover much of the young forest landscape indicating that extensive and severe reburning may be possible, especially under severe fire weather conditions. Post-fire stand structure, especially live lodgepole pine stem density, was the single greatest predictor of canopy and fine fuel loads, but pre-fire successional stage was the most important predictor of large woody surface fuels. Stands in this study have passed the 10–18-yr period where severe reburning potential is reduced (Parks et al. 2014, 2015, Harvey et al. 2016); however, some stands with low post-fire regeneration have low fuel loads and may provide some resistance to subsequent fire.

A surprising finding was the high variability in fuel conditions across the post-fire landscape suggesting substantial spatial variability in potential fire behavior and severity. Fine surface fuel loads in these 24-yr old forests were generally less than those reported in mature lodgepole pine forests, but most stands contained sufficient litter and

Table 3. Predictive linear models illustrating the effects of post-fire stand structure, topo-climatic factors, and pre-fire successional stage on dead surface fuel loads in 24-yr-old lodgepole pine stands.

Dead surface fuels	df	R^2	Parameter	ß	SE	t	P
$\log_{10}$ (litter fuels; Mg/ha)	78	0.43	intercept	0.41	0.57	0.72	0.474
			1,000-h fuel biomass	0.003	0.000	3.11	0.003
			$\log_{10}$(lodgepole pine density)	0.31	0.05	6.31	<0.001
			mean annual precipitation	−0.002	0.001	−2.70	0.009
$\log_{10}$ (1-h fuels) (Mg/ha)	77	0.49	intercept	−2.19	0.31	−7.11	<0.001
			mean crown base height	0.71	0.09	7.75	<0.001
			mean annual precipitation	0.001	0.000	2.83	0.006
			topographic wetness index	0.02	0.009	2.15	0.034
			slope	0.02	0.007	2.08	0.041
10-h fuels (Mg/ha)	79	0.08	intercept	0.89	0.55	1.63	0.106
			$\log_{10}$(lodgepole pine density)	0.36	0.14	2.55	0.013
			mean annual temperature	−0.03	0.01	−1.96	0.053
$\log_{10}$ (100-h fuels; Mg/ha)	78	0.22	intercept	0.67	0.02	28.91	<0.001
			early pre-fire successional stage (LP1)	−0.15	0.06	−2.63	0.012
			middle pre-fire successional stage (LP2)	0.17	0.04	4.66	<0.001
			late pre-fire successional stage (LP3/4)	−0.01	0.03	−0.49	0.62
$\log_{10}$ (sound-1,000-h fuels; Mg/ha)	77	0.24	intercept	0.65	0.52	1.25	0.216
			early pre-fire successional stage (LP1)	−0.32	0.09	−3.72	<0.001
			middle pre-fire successional stage (LP2)	0.07	0.06	1.10	0.275
			late pre-fire successional stage (LP3/4)	0.17	0.05	3.66	<0.001
			elevation	0.001	0.000	2.08	0.040
$\log_{10}$ (rotten-1,000-h fuels; Mg/ha)	77	0.30	intercept	2.62	0.44	5.91	<0.001
			early pre-fire successional stage (LP1)	0.22	0.12	1.82	0.073
			middle pre-fire successional stage (LP2)	−0.38	0.08	−4.71	<0.001
			late pre-fire successional stage (LP3/4)	0.004	0.07	0.07	0.95
			mean annual precipitation	−0.002	0.000	−2.79	0.007
Total surface fuels (Mg/ha)	75	0.31	intercept	379.68	59.70	6.36	<0.001
			early pre-fire successional stage (LP1)	−24.23	14.40	−1.68	0.10
			middle pre-fire successional stage (LP2)	−6.89	10.81	−0.64	0.53
			late pre-fire successional stage (LP3/4)	18.99	8.01	2.37	0.02
			mean annual precipitation	−0.042	0.10	−4.03	<0.001
			mean annual temperature	−2.42	0.72	−3.34	0.001
			aspect	−15.94	6.28	−2.54	0.013

1-h fuels to support rapid surface fire spread. Coarse fuel loads were similar or higher than values observed in mature lodgepole pine forests after other disturbance types (e.g., bark beetles and blowdown; Veblen 2000, Woodall and Nagel 2007, Kulakowski and Veblen 2007), and stands with abundant coarse fuels are especially susceptible to prolonged smoldering with high biomass consumption and heat release (Byram 1959, Rothermel 1972, Scott and Reinhardt 2001, Sikkink and Keane 2012). Canopy fuel loads, and particularly canopy bulk density, attained or exceeded values reported in mature lodgepole pine forests, and ubiquitous low canopy base heights indicate that young stands may be susceptible to crown fire initiation and many stands can support active and independent crown fire spread. Fuel conditions in most stands suggest that fire may be difficult to control, particularly in places where fires must be suppressed (e.g., near infrastructure).

Surface fuel loads in developing lodgepole pine forests differed from those found in other young and mature forest types. Litter and 1-h fuels were about half that found in 1-, 3-, and 19-yr-old western hemlock/Douglas-fir stands in Washington, USA (Agee and Huff 1987), and 2-, 27-, and 30-yr-old subalpine fir stands in Montana, USA (Fahnestock 1976), similar to Swedish Scots pine/Norway spruce/birch boreal forests <20-yr old (Schimmel and Granström 1997) and mixed-conifer stands in the Cascade range, USA, measured 1 yr after high-severity fire (Hudec and Peterson 2012), but greater than a 48-yr-old Jack pine (*Pinus banksiana*) stand that originated from fire in Ontario, Canada (Stocks 1987). The complete absence of duff in our study also differs from these studies but reflects results from other studies that documented little duff accumulation in lodgepole pine stands across Yellowstone National Park (Romme 1982, Litton et al. 2004, Kashian

TABLE 4. Predictive linear models illustrating the effects of post-fire stand structure, topo-climatic factors, and pre-fire successional stage on live surface and canopy fuel loads in 24-yr-old lodgepole pine stands.

	df	R^2	Parameter	ß	SE	t	P
Live surface fuels							
Live herbaceous fuels (Mg/ha)	77	0.25	intercept	2.09	0.34	6.25	<0.001
			$\log_{10}$(lodgepole pine density)	−0.24	0.09	−2.78	0.007
			rhyolite–till	−0.14	0.19	−0.73	0.470
			rhyolite–glacial	−0.12	0.16	−0.70	0.484
			rhyolite–low base saturation	−0.23	0.11	−2.12	0.037
$\log_{10}$(live shrub fuels; Mg/ha)	78	0.25	intercept	−8.75	1.63	−5.36	<0.001
			$\log_{10}$(lodgepole pine density)	0.35	0.10	3.60	<0.001
			elevation	0.002	0.01	4.27	<0.001
			fire severity (dNBR)	0.001	0.01	2.70	0.009
Live canopy fuels							
$\log_{10}$(available canopy fuels; Mg/ha)	81	0.78	intercept	−1.27	0.12	−10.70	<0.001
			$\log_{10}$(lodgepole pine density)	0.54	0.03	16.95	<0.001
$\log_{10}$(mean crown base HT) (Mg/ha)	79	0.66	intercept	−1.16	0.08	−15.12	<0.001
			$\log_{10}$(lodgepole pine density)	0.24	0.02	12.01	<0.001
			slope	−0.01	0.004	−2.24	0.028
$\log_{10}$(canopy bulk density; kg/m^3)	81	0.91	intercept	−3.39	0.09	−39.62	<0.001
			$\log_{10}$(lodgepole pine density)	0.70	0.02	29.98	<0.001

et al. 2013). Previous studies in mature, Rocky Mountain lodgepole pine forests report similar litter, 2–4 times higher 1-h fuel loads, similar 10-h fuel loads, and 10–20 times lower 100-h and 1,000-h fuel loads than were found in this study (Lawson 1973, Alexander 1979, Lotan et al. 1985, Battaglia et al. 2010). Studies in mature, Rocky Mountain ponderosa pine (*Pinus ponderosa*) forests documented similar litter loads, two times higher 1-h fuel loads, 2–10 times higher 10-h fuel loads, and similar but low 100-h and 1,000-h fuel loads than were observed in these post-1988 lodgepole pine forests (Mason et al. 2007, Klutsch et al. 2009, Battaglia et al. 2010). Studies in mixed conifer forests (i.e., *Pinus* spp./*Abies* spp./*Calocedrus* spp./*Quercus* spp.) in the Sierra and Cascade Mountain ranges observed 1–10 times greater litter and 1-h fuel loads, 1–5 time higher 10-h fuel loads, and 100-h and 1,000-h fuel loads similar to loads observed this study (Schmidt et al. 2008, Van de Water and North 2011, Hudec and Peterson 2012, Pierce et al. 2012, Banwell and Varner 2014, Lydersen et al. 2015).

Our hypothesis that the legacy of pre-fire forest structure would greatly influence 100-h and 1,000-h fuel loads and the proportion of rotten and sound logs was supported by the distribution of log sizes stratified by successional stage (Fig. 2) but only weakly supported by our regression models (Table 3). Pre-1988 successional stages, as implemented in this study, were derived from a geospatial cover type map produced via aerial photo classification and field verification (Despain 1990, NPS-YELL 1990). We expect that regression results involving pre-fire successional stage could be improved if more detailed pre-fire stand structure measurements had been available. Still, our findings highlight the influence of pre-fire successional stage on coarse fuel loads and indicate that all old coniferous forests, or forests that otherwise had large trees at the time of the fire, will likely have more coarse wood after fire, as has been demonstrated after short-interval fire in the Pacific Northwest (Donato et al. 2016). Subsequent fires that burn in stands that had large tress prior to the first fire will likely have greater biomass consumption, flame residence time, heat release, and smolder for longer than will forests of small trees or sparse trees prior to the initial fire.

Lodgepole pine forests are unique in being able to develop an enormous canopy seedbank that facilitates abundant seedling establishment after fire (Clements 1910, Lotan et al. 1985). Twenty-four years post-fire, canopy fuel loads attained or exceeded values reported in other young and mature forest types. On average, canopy fuel loads in these stands were ~5 times higher than values observed in 19-year-old western hemlock/Douglas-fir forests in Washington, USA (Agee and Huff 1987), but similar to subalpine fir stands 27- and 30-year post-fire in Montana, USA (Fahnestock 1976). Approximately 45% of stands in this study have available canopy fuel loads greater than 10 Mg/ha and the highest available canopy fuel load (48 Mg/ha) in our study is greater than the maximum value found in mature lodgepole pine forests (Cruz et al. 2003, Reinhardt et al. 2006, Simard et al. 2011). Canopy bulk density was ~2–5 times greater than values found in mature lodgepole pine, ponderosa pine, and mixed conifer stands in Colorado, Idaho, and Montana, USA (Cruz et al. 2003, Fulé et al. 2004, Reinhardt et al. 2006, Simard et al. 2011, Roccaforte et al. 2015), but similar to untreated ponderosa pine and mixed conifer stands affected by fire suppression in Arizona, USA (Cruz et al. 2003, Hall and Burke 2006, Reinhardt et al. 2006, Mason et al. 2007). Results from this study highlight the link between post-fire tree regeneration and the development of canopy and fine surface fuels, suggesting that stands with the highest rates of light capture and biomass production also have the highest canopy and fine surface fuel loads. In forest types lacking such a seedbank, we would expect slower canopy fuel development and reduced crown fire potential after a quarter century of post-fire recovery.

Just how much of the landscape is now vulnerable to high-severity re-burning and/or active crown fire spread?

TABLE 5. The relative contribution of predictor variables to the models explanatory power with significant differences evaluated using bootstrapped confidence intervals.

Fuel class response variables	Stand structure and environmental predictor variables												Interpretation
	Live stand density	Crown base height	1,000-h fuel load	Pre-fire succes-sional stage	Substrate	Mean annual precipita-tion	Mean annual tempera-ture	Compound topographic index	Elevation	Slope	1988 dNBR	Aspect	
Dead surface fuels													
Litter	0.66[a] (+)	–	0.13[b] (+)	–	–	0.21[a,b] (−)	–	–	–	–	–	–	Litter biomass varied positively with post-fire stand attributes linked to litter deposition (i.e., stand density) and 1,000-h fuel loads, which promote drier soil conditions found on site with high numbers of logs (Remsburg and Turner 2006). Annual precipitation was negatively related to litter biomass possibly due to suppressed decomposition rates.
1-h	–	0.78[a] (+)	–	–	–	0.09[b] (+)		0.02[b] (+)	–	0.01[b] (+)	–	–	1-h fuels increased with post-fire stand attributes (i.e., canopy base height and stand density) related to lower branch pruning and with moisture availability.
10-h	0.70[a] (+)	–	–	–	–	–	0.30[a] (−)						10-h fuels were weakly predicted by post-fire stand attributes linked to litter deposition (i.e., stand density) and mean annual temperature.
1,00-h	–	–	–	1.0 (+,−)	–	–	–	–	–	–	–	–	100-h fuels varied negatively with early and late pre-fire successional stages and positively with middle pre-fire successional stages. These relationships are believed to stem from the sizes of pre-fire trees and logs.
1,000-h, sound	–	–	–	0.99[a] (+,−)	–	–	–	–	0.01[b] (+)	–	–	–	Sound 1,000-h fuels varied negatively with early and middle pre-fire successional stages and positively with late pre-fire successional stages. The greatest sound 1,000-h fuel loads occurred on sites with large size classes of pre-fire trees and logs.

(*Continued*)

Table 5. Continued.

Fuel class response variables	Stand structure and environmental predictor variables												Interpretation
	Live stand density	Crown base height	1,000-h fuel load	Pre-fire successional stage	Substrate	Mean annual precipitation	Mean annual temperature	Compound topographic index	Elevation	Slope	1988 dNBR	Aspect	
1,000-h, rotten	–	–	–	0.99[a] (−)	–	–	–	–	0.01[b] (−)	–	–	–	Rotten 1,000-h fuels varied positively with early pre-fire successional stage and negatively with middle and late pre-fire successional stages. Rotten 1,000-h fuels are highest on sites with small size classes of pre-fire trees and logs.
Total surface	–	–	–	0.28[a] (+)	–	0.34[a] (−)	0.27[a] (−)	–	–	–	–	0.12[a] (−)	Total surface fuel load varied positively with pre-fire successional stage and negatively with mean annual temperature, mean annual precipitation, and aspect.
Live surface fuels													
Live herbaceous	0.38[a] (−)	–	–	–	0.61[a] (−)	–	–	–	–	–	–	–	Herbaceous biomass declined with post-fire stand basal area (i.e., restricted light and soil resources) and increased with substrate quality.
Live shrub	0.25[a] (+)	–	–	–	–	–	–	–	–	0.50[a] (+)	0.25[a] (+)	–	Shrub biomass varied positively with post-fire stand density (i.e., restricted light and soil resources) and elevation.
Live canopy fuels													
Available canopy fuel	1.0 (+)	–	–	–	–	–	–	–	–	–	–	–	Available canopy fuel load varied positively with post-fire stand density.
Crown base height	0.96[a] (+)	–	–	–	–	–	–	–	–	0.04[b] (−)	–	–	Canopy bulk density increased with post-fire stand density. High density stands were found to have the greatest foliar biomass and the lowest canopy length.
Canopy bulk density	1.0 (+)	–	–	–	–	–	–	–	–	–	–	–	Canopy base height increased with post-fire stand density as a result of density-dependent lower branch pruning.

Note: Each predictor's effect on the response is denoted using positive (+) and negative signs (−).

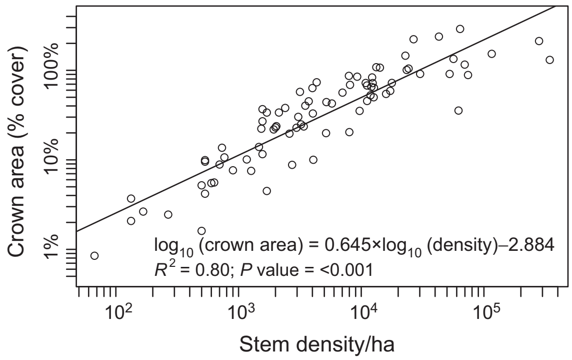

FIG. 3. The relationship between stem density and percent crown area. Crown area is defined as the percentage of ground area covered by tree crowns assuming circular crowns and even tree spacing.

If we assume that sampled stands represent the proportion of a given fuel characteristic across the burned landscape, then approximately 76% of the post-1988 fire landscape is susceptible to high-severity surface fire and 63% is capable of active or independent crown fire spread. In a simulation study investigating surface-fire severity using the First Order Fire Effects Model (FOFEM), Sikkink and Keane (2012) found that dry coarse fuel loads above the 60 Mg/ha threshold resulted in a mean fire line intensity greater than 120 Kw/m^2, a mean fire residence time greater than 2 h, and soil temperatures greater than 60°C to a >6 cm depth. Using the mean intensity reported by Sikkink and Keane (2012), canopy fuels would be capable of igniting at 0.8-m canopy base height and 100% live foliar moisture content using Van Wagner's (1977) crown fire initiation equation, encompassing approximately 90% of the stands sampled in this study.

Changes in surface fuels between 1996 and 2012 showed that fuel deposition from growing young trees and falling fire-killed trees were the dominant factors shaping surface woody fuels during the first 24 years of forest development (Table 6). Though our sample size is limited, these plots bridged the 10–18 years post-fire period beyond which fire potential and severity are not reduced by previous fire (Parks et al. 2014, 2015, Collins et al. 2015, Harvey et al. 2016). Fuel loads increased by 175–430% during this window of time (1-h fuels excepted; Table 6) and within-plot variability declined, indicating a transition from patchy to more spatially continuous fuel beds (Fig. 5). Severe surface fire potential increased dramatically during this period due to snag fall from 10% of stands in 1996 to 90% of stands in 2012. Overall, the rapid increase in surface fuel loads is consistent with Kashian et al.'s (2013) finding that ~50% of maximum needle and woody litter is recovered in the first 25 years after fire. Delayed post-fire snag fall likely accounts for increases in 100-h and 1,000-h fuel loads since large fuel classes are subject to slow biomass turnover rates and can take ~125 years to decompose completely (Kashian et al. 2013). The lack of change in 1-h fuels was not surprising given the high biomass turnover rate for small diameter wood observed by Simard et al. (2012).

CONCLUSION

For land management agencies to develop informed adaptation and mitigation strategies to attenuate adverse impacts of increased fire activity to human life, infrastructure, and ecological services, quantitative data on the variability and dynamics of fuel beds in young forests will become increasingly important. On lands implementing passive management strategies such as wildland fire use, management personnel should acknowledge that young

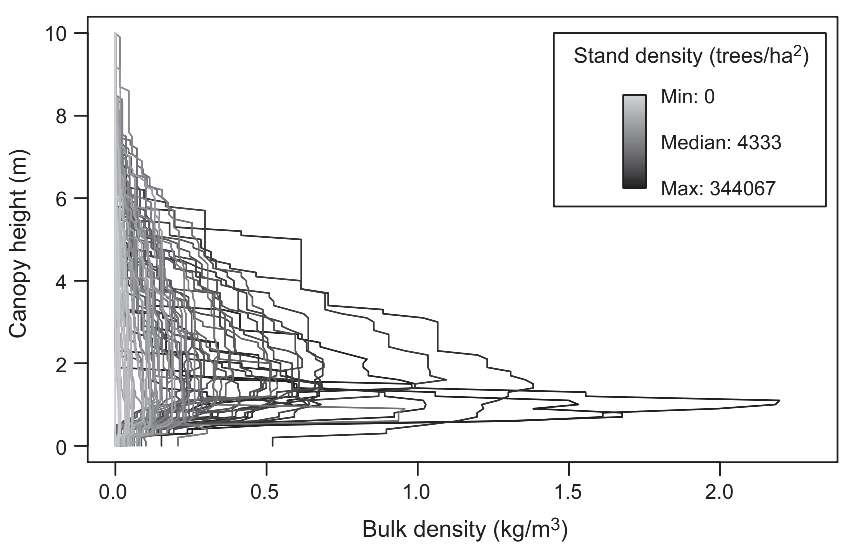

FIG. 4. Vertical variation in canopy bulk density by stand density. Canopy bulk density was estimated for 0.1-m vertical strata by summing available canopy fuel load within each strata then dividing by the volume of each layer (plot area × strata depth) for each plot and smoothed for plotting using a 1-m running mean.

Table 6. Changes in surface fuel loads between 1996 and 2012 in 10 remeasured plots.

Surface fuel type	1996 (Mg/ha)	2012 (Mg/ha)	t	P (two-tailed)	Ratio of change (2012/1996)
1-h fuels	0.13 ± 0.02 [0.07, 0.22]	0.14 ± 0.02 [0.06, 0.26]	−0.34	0.744	1.07 ± 0.22
10-h fuels	0.98 ± 0.15 [0.00, 1.92]	2.37 ± 0.16 [1.52, 3.27]	−5.96	<0.001	2.41 ± 0.43
100-h fuels	2.92 ± 0.39 [0.40, 4.42]	5.07 ± 0.46 [1.97, 6.66]	−3.82	0.004	1.74 ± 0.26
1,000-h fuels, sound	26.85 ± 4.14 [11.40, 50.16]	60.61 ± 7.42 [25.54, 95.78]	−3.99	0.003	2.26 ± 0.44
1,000-h fuels, rotten	11.07 ± 2.08 [1.72, 19.71]	47.66 ± 6.79 [23.15,79.20]	−6.18	<0.001	4.31 ± 0.70
Total fuel load	41.95 ± 5.16 [24.01, 74.92]	115.84 ± 9.61 [60.17, 160.54]	−7.49	<0.001	2.76 ± 0.37

Notes: Mean ± 1 SE and the range of observations in each time period. Statistics reflect paired t tests.

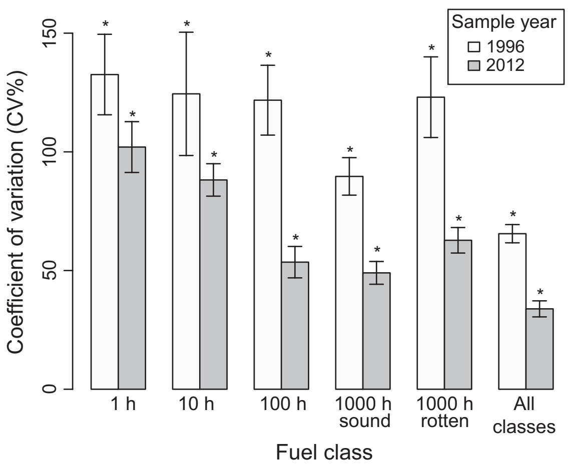

Fig. 5. Within-plot coefficient of variation for surface fuel loads in 1996 and 2012. Asterisks indicate significant differences between years using a two-sided, paired t test ($\alpha = 0.05$).

forests may have heavy fuel loads, like those in many of the stands in this study, capable of sustaining stand-replacing fire. On lands under active management, post-fire salvage logging may be implemented to reduce long-term coarse fuel loads and lessen resistance to control.

In conclusion, the tremendous variation in fuel loads across the post-1988 fire landscape suggest that stand age alone is a poor surrogate for predicting fuel conditions in young lodgepole pine stands that regenerated naturally from stand-replacing fire. Surprisingly, these stands have already developed fuel conditions that are likely to sustain reburning. Most post-1988 fire lodgepole pine forests can likely sustain high-severity surface fire and active crown fire, although we anticipate that fire behavior and effects will vary spatially across the landscape. In the future, fire rotations in Yellowstone National Park are predicted to be shorter than were typical historically (Westerling et al. 2011), and this prediction cannot be ruled out by a lack of fuels to carry repeated fires at intervals of a few decades.

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.1412/full

Data Availability

Data associated with this paper have been deposited in Dryad: http://dx.doi.org/10.5061/dryad.3b15s (Nelson et al. 2016).